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Lead in Soil: Issues and Guidelines

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Modelling the Blood Lead – Soil Lead Relationship*

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Abstract

The relationship between blood lead and lead uptake from the environment has been described by cross-sectional models, by causal models based on longitudinal data, and by predictive models based on hypotheses about lead uptake and biokinetics in children. The empirical blood lead versus soil lead linear slope in many studies is about 2 μg/dL per 1,000 μg/g soil lead. The uptake/biokinetic models are currently being validated and give good estimates of mean blood lead in children. Biological and statistical factors may cause nonlinearity of the relationship at blood lead levels above 25 μg/dL.

Introduction

Elevated lead levels in surface soil and environmental dust cause elevated blood lead concentrations in many young children who spend extended periods of time living or playing near the contaminated soil.

The mechanisms are understood in general. Surface dust may be ingested by normal hand-to-mouth activity, especially in one- to three-year-old children (Duggan *et al.*, 1985). Some may consume excessively large quantities of soil and other non-food items, a condition known as *pica*. Contaminated soil may become accessible by weathering, or by humans or animals digging up the soil. Soil particles can become airborne and be inhaled, can contribute to internal household dust by atmospheric entry or can be tracked into the house on shoes, clothing and pets. More work on the quantification of these processes is needed.

*This paper represents the views of the authors and does not represent the view of the Battelle Memorial Institute or the United States Environmental Protection Agency.

At present we have three major approaches to modelling the relationship between soil lead and blood lead:

(1) Descriptive cross-sectional models.
(2) Descriptive causal longitudinal models.
(3) Predictive longitudinal simulation models based on multimedia lead uptake and biokinetics.

The basic issues are described in EPA's Air Quality Criteria for Lead (US EPA, 1986a). Lead exposure arises from multiple sources (air, food, water, soil, dusts) and multiple pathways (inhalation, ingestion). These must be controlled, either by experimental design or by observation and covariate adjustment in order to obtain an unbiased estimate of the soil lead contribution.

Descriptive Cross-sectional Models

Most of the analyses of soil lead have involved cross–sectional studies of children in several locations with different air lead and soil lead exposures, and all blood lead (PbB) concentrations taken at a single point in time at each location. Listed below are those we believe to be of greatest relevance, many conducted near point sources, such as lead smelters, which have substantial elevations of soil lead in their vicinity. These studies are:

(1) Omaha, Nebraska (Angle *et al.*, 1984)
(2) New Haven, Connecticut (Stark *et al.*, 1982; EPA, 1986a)
(3) Kellogg/Silver Valley, Idaho (Yankel *et al.*, 1977; EPA, 1986a)
(4) Charleston, South Carolina (Galke *et al.*, 1975)
(5) Towns in England (Barltrop, 1975; EPA, 1986a)
(6) Towns in British Columbia (Neri *et al.*, 1978; EPA, 1986a)
(7) East Helena, Montana (CDC–MDHES,1983; Johnson and Wijnberg, 1988).

Serious problems arise in comparing these studies. Different sets of covariates were used in different studies (including dust lead and air lead levels that were often highly correlated with soil lead levels), populations of children with different age distributions were used, and different soil lead sampling protocols were used. The results are shown in Table 1, an extension of Table 11–63 in EPA's Air Lead Criteria Document (US EPA, 1986a). Many of these data are re–analyzed by EPA, using the fact that blood lead levels are usually close to being lognormally distributed, even among individuals with the same lead exposure. The EPA fitted models are in the form

$$\log(\text{PbB}) = \log[\text{slope} \times (\text{soil lead}) + \text{other terms}] \quad (1)$$

This model assumes that the effects of separate lead exposure pathways contribute additively to blood lead geometric means or arithmetic means.

These assumptions are known to be wrong for blood lead levels much above 25–30 μg/dL in children, and 35–40 μg/dL in adults. Nonlinear

Table 1. *Estimates of the contribution of soil lead to blood lead. (From Table 11–63, Air Quality Criteria for Lead, EPA, 1986.)*

Range of soil lead values (μg/g)	*Depth sample (cm)*	*Estimated slope (×10³)*	*Sample size*	*R^2*	*Study*
16–4,792	5.0	6.8	1,075	0.198	Angle and McIntire (1982) study of children in Omaha, NE.
30–7,000 (age 0–1)	1.3	2.2	153	0.289	Stark *et al.* (1982) study of children
30–7,600 (age 2–3)		2.0	334	0.300	in New Haven, CT.
50–24,600 (age 2–3)	1.9	1.1	860	0.662	Yankel *et al.* (1977) study of children in Kellogg, ID.
9–7,890	5.0	1.5	194	0.386	Galke *et al.* (1975) study of children in Charleston, SC.
420–13,969 (group means)	5.0	0.6	82	NA	Barltrop *et al.* (1975) study of children in England.
225–1,800 (group means) (age 1–2)	NA	7.6	87	NA	Neri *et al.* (1978) study of children in British Columbia.
225–1,800 (group means) (age 2–3)	NA	4.6	103	NA	" "
55–3,415	2.0	1.4	186	0.369	CDC–MDHES (1983) study of children in East Helena, MT. (Johnson and Wijnberg, 1988)

NA = information not available.

processes in lead absorption (*e.g.* a saturable gut absorption pathway) leads to non–additivity of uptakes, while nonlinearity in lead distribution to blood and tissues causes nonlinearity of the response function. This is discussed later.

Note that we have included some recent unpublished results (Johnson and Wijnberg, 1988) based on the 1983 CDC–Montana Dept. Health and Environmental Science study of young children in East Helena, Montana. The blood lead–soil lead regression coefficient or "slope" in that re-analysis was also adjusted for numerous covariates, including children's age, air and dust lead, poor quality lead painted housing, smoking, and secondary occupational exposure.

Results are generally consistent, given the uneven quality of the data. This suite of studies suggests that children 1–3 years old will have an extra

lead burden due to soil lead exposure in their vicinity, the typical excess amounting to about 2 μg/dL per 1,000 μg/g soil lead (median of study values: slope range 0.6–7.6). The contribution of household dust lead was partially factored out of most of the US EPA analyses. In many of the studies, the dust lead slope is similar to the soil lead slope, suggesting that soil lead is a substantial contributor to indoor dust lead so that soil contributes both directly and indirectly to blood lead. This hypothesis has been explored in detail in the longitudinal studies described in the next section.

Descriptive Causal Models Using Longitudinal Data

If data on blood lead, soil lead and other exposure variables have been obtained over time, then it may be possible to infer cause-effect responses of blood lead to earlier lead exposures. Good longitudinal data on blood lead and developmental indicators were obtained on Cincinnati (Bornschein *et al.*, 1985, 1986) and Boston children (Rabinowitz *et al.*, 1985; Bellinger *et al.*, 1987). The Cincinnati data set was analyzed by *structural equation* methods (Bentler, 1980) for relationships between exterior surface scrapings, dust lead, and blood lead. Unfortunately, the results are expressed as a linear equation in logarithms of the environmental variables, and the standardized regression coefficients are not directly interpretable as overall slopes for soil or dust lead given the published information (Bornschein *et al.*, 1986).

The analyses of the Boston data used a somewhat different technique, random effects models for longitudinal data (Ware, 1985). Unfortunately, soil lead data were not sampled longitudinally. There were repeat measurements for blood, dust, air and water lead. The 18-month blood lead regression model reported in (Rabinowitz *et al.*, 1985) used the logarithm of the yard soil lead measurement as a predictor, so that the reported slope is not directly comparable with those reported in the section on descriptive cross-sectional models. However, the slopes at several concentrations are similar to those in Table 1, *i.e.* 8 μg/dL per 1,000 μg/g soil lead when soil lead = 100 μg/g, 1.6 μg/dL per 1,000 μg/g soil lead when soil lead = 500 μg/g, and 0.8 μg/dL per 1,000 μg/g when soil lead = 1,000 μg/g (the average soil lead for the Boston children with "high" blood lead was 1,011 μg/g, and 380 μg/g for those with "low" blood lead). Nonlinearity in the model is evident.

Most of the published analyses have also evaluated the response of blood lead to dust lead and hand lead changes. It is clear that this methodology is appropriate to estimating the direct and indirect influence of soil lead changes over time that would be expected during soil lead abatement programmes. There may be some methodological problems in dealing with intrinsically nonlinear equations such as equation (1) but this methodology is evidently appropriate to identifying the interrelationships of changes in soil lead, air and dust lead on blood lead levels in growing children.

Predictive Simulation Models Based on Lead Uptake and Biokinetic

OAQPS uptake/biokinetic model

While these observational models are highly useful, they often fall short of providing the scientific information necessary for regulatory purposes. We need to estimate the changes, over time, of blood lead and other biological responses to changes in environmental lead in a rapidly developing young child. We need to estimate the effects of hypothetical soil abatement strategies on normal children and on high-risk subpopulations such as children with *pica* and iron-deficient diets. Thus EPA's Office of Air Quality Planning and Standards (OAQPS) has developed an integrated lead uptake/biokinetic model.

A general description of the model is provided in the draft OAQPS Staff Paper (US EPA, 1988b; ATSDR, 1988). Average daily uptake for young children is calculated under conditions specified in terms of ambient air lead levels, soil and dust lead levels that correspond to both historical and current atmospheric lead emissions, and dietary lead levels from both water and food. Water lead is non-atmospheric in origin and results mainly from corrosion of materials in distribution and residential plumbing systems. Lead in food comes from atmospheric lead deposition, metallic contamination, or from natural sources. Additional exposures to paint lead or secondary occupational sources (*e.g.* dust tracked into homes by workers) can be added. Blood lead levels (PbBs) in populations of young children, who are the most exposed and highly susceptible to lead, are estimated over time as a function of total daily lead uptake using a biokinetic compartmental model developed by Harley and Kneip (1985) of New York University.

Parameters such as indoor air lead exposure, time spent indoors *versus* outdoors, absorption rates through the lung or gastrointestinal tract, and amount of dirt children typically ingested through hand-to-mouth activity, are estimated from available data in the literature, as summarized in the air quality criteria document and the staff paper. In some cases, relevant data have become available since publication of the 1986 criteria document. For example, new estimates on dirt ingestion rates in children, discussed elsewhere in this conference, fill an especially critical data gap along with additional information on soil and dust lead levels in relation to air lead concentrations. This new information has been incorporated in estimating various exposure parameters summarized here and has been added to the final staff paper assessment.

Validation of the model is currently in progress using several data sets. The data set considered first is based on a 1983 study in which the Montana Department of Health and Environmental Sciences (MDHES) and the Centers for Disease Control (CDC), in cooperation with EPA, measured blood lead levels in approximately 300 children ages 1 to 5 living around a lead smelter in East Helena. The lead content of soil and dusts around and in individual homes was measured. Air-borne lead was measured before and during the survey at 8 sites. Three study areas were designated according to their distances from the smelter: Area 1, within 1.6 km of the smelter; Area 2, 1.6–3.7 km from the smelter; Area 3, more than 8 km

from the smelter. Unfortunately, only one site, about 0.4 km from the smelter, provided sufficiently complete air quality data over the period 1978–1983 that could be considered representative of population exposure in vicinity of the monitor. However, air lead data were available from other monitors, at various times, in 10 other locations around East Helena.

In order to estimate 1983 PbBs in the East Helena children so they could be matched against the measured PbBs, exposure profiles were generated extending as far back as 1978, depending on the children's age in 1983 and their residential location. A five-year description of monthly ambient air lead concentrations throughout the study area was generated in two ways: (1) use of the Industrial Source Complex-Long Term (ISC–LT) dispersion model based on data on air lead concentrations, smelter emissions, and local meteorology, and which accounted for dry atmospheric deposition; (2) empirical smooth interpolation of observed concentrations (Marcus 1988a). Area and year-specific estimates of background lead contributions, (*i.e.* mobile sources, re-entrained soil, local minor point sources) were also included.

Table 2 presents values used for the different exposure parameters in the model for a 2-year old child as an illustration. Given the modelled ambient air lead levels, lead exposure *via* inhalation for each child is calculated based on estimated indoor air lead concentration, age-specific indoor *versus* outdoor time partitioning and age-specific ventilation and respiratory deposition/absorption rates.

Using methods adopted in the Air Quality Criteria document for Lead, year- and age-specific dietary lead exposure was estimated from FDA data

Table 2. *Input parameters for uptake/biokinetic model for 2-year olds living near East Helena smelter in 1983.*

Time spent indoors	2–4 hours/day
Volume of air respired	4–5 m^3/day
Deposition/absorption of inhale d Pb in lung	25–45%
Dietary lead consumption	
a) from solder or other metals	7.4 μg/day
b) atmospheric lead	9.2
c) natural lead, indirect atmospheric, undetermined	4.4
Absorbtion of dietary Pb in gut	30–40%
Amount of dirt ingested	0.085–0.13 g/day
Absorption of "dirt" Pb in gut	20–30%

Values represent averages and differ for different age groups. Dietary lead consumption estimates vary for different years because of changes in canning technology and atmospheric deposition. Time-weighted averages of indoor and ambient air concentrations amd indoor dust and outdoor surface soil/dust lead concentrations are key inputs. Dust and soil lead concentrations were derived from actual measurements, or estimated from generic relationships depending on the model run. Air lead concentrations were derived from monitoring data and dispersion model predictions. Indoor air lead levels were estimated near the smelter to be 30% of ambient air lead levels.

on food lead content (*i.e.* market basket surveys) and dietary patterns throughout childhood. The contribution of drinking water to dietary lead exposure was estimated for an average tap water concentration of 12 μg/L which was reported for the State of Montana in 1983. Age-specific data on lead absorption through the gastrointestinal tract were applied to calculate dietary lead uptake.

Exposure to lead from contaminated soil and dust is estimated in the integrated uptake model by: 1) calculating a time weighted average concentration of soil/dust lead to which children would be exposed (based on indoor and outdoor dust/soil lead levels and age-specific indoor *versus* outdoor time partitioning); 2) assuming typical age-specific rates of daily dirt ingestion through "normal" hand-to-mouth activity (excluding abnormal non-food eating habits, or pica); and 3) applying a gut absorption rate for lead in "dirt". Indoor dust lead and outdoor surface soil lead concentrations were estimated using home-specific measurements reported by MDHES and CDC. Dirt consumption rates are derived for the model from published studies using tracer element analysis (Binder *et al.*, 1986; Clausing *et al.*, 1987) and informal discussions with experts in CDC as elsewhere who have studied this route of exposure.

In estimating the rate by which ingested lead is absorbed, US EPA (1986, 1988) identifies many factors that had to be considered, among them person's age, physiological status, medium for the lead (*e.g.* food *versus* paint or "dirt"), coincident ingestion of other elements in food (*e.g.* calcium) which compete with lead for intestinal absorption sites, and the non-linear relationship between absorption and ingestion rates (*e.g.* lead absorption declines with increased intake thereby serving as a protective mechanism for the body). This latter phenomenon may have been present in East Helena where the closest fits between observed and predicted blood lead levels were obtained in exploratory analysis when an absorption factor of 20% was used for Area 1 children exposed to high soil and dust lead levels and 30% was applied to children living more distant from the smelter. These differences may also be attributable to physical or chemical differences in bioavailability of near *versus* distant smelter particulate emissions.

Average blood lead levels are calculated from the insertion of daily uptake estimates into the biokinetic model of lead metabolism for children developed by Harley and Kneip (1985).

This model is based on dynamic blood measurements and steady state blood and organ lead measurements in young baboons chronically exposed under controlled conditions and was subsequently adjusted for human metabolism and growth patterns in children for each organ. The biokinetic model was successfully validated against human autopsy data and is consistent with dietary experiments in which blood and skeletal lead accumulation was measured in infants and adults.

Results for East Helena Children

Age-specific estimates of daily lead uptake through childhood were entered into the Harley and Kneip biokinetic model described previously.

Table 3. *Comparison of East Helena blood lead distribution parameters with uptake/biokinetic simulation model.**

	Geometric mean blood lead (μg/dL)	*Geometric standard deviation*
Area 1 (<1 Mile)		
Observed	11.29	1.58
Model	12.18	1.61
Area 2 (1–2.25 Miles)		
Observed	8.62	1.55
Model	8.31	1.37

*Run 1B parameters.

Blood lead levels estimated for 1983 were then compared with the individual MDHES/CDC measurements. Table 3 summarizes the results from a recent run using the interpolated air leads (Johnson, 1988).

The results are generally satisfactory for predicting geometric means, but the predicted blood lead levels appear to have a less dispersed distribution (smaller geometric standard deviation) than the observed blood leads. This is undoubtedly due to variations in biological parameters and in uptake from other sources that are not included in the deterministic simulation model. The model should be regarded as valid for estimating mean responses. Accurate estimation of the higher percentiles of the blood lead distribution requires application of an empirically-derived GSD to the geometric mean. For example, unadjusted GSDs from NHANES II or point source studies may be used.

Application of uptake/biokinetic model to other data sets

There are other published data to test the uptake/biokinetic model at other US locations and times. One data set was collected in Omaha, Nebraska, from 1971 to 1977 (Angle and McIntire, 1979; Angle *et al.*, 1984). Another data set was collected throughout Silver Valley, Idaho, in 1974–1975 and described in Yankel *et al.* (1977), and Walter *et al.* (1980) with additional information given in the draft OAQPS Staff Paper and in the lead Criteria Document.

The Omaha study determined blood lead for 242 children ages 1 to 5 years and 832 children ages 6 to 18 years. The children were at 3 locations, denoted C (commercial area near a small battery plant), M (mixed residential-commercial), and S (suburban). Air lead, soil lead, and house dust lead measured in each area are incorporated into the model to estimate average daily lead uptake levels. These data are much less satisfactory for testing the model than the East Helena data because the soil and dust lead measurements are not specific to each household where blood leads were taken. Furthermore, the study was not designed for controlled geographic comparisons given the significant difference between areas in SES, housing, and racial makeup. Age-specific dietary lead intakes were calculated based on 1970s data from the FDA as re-analyzed in the lead Criteria Document. In Table 4, the observed geometric mean

and 1975. PbB levels were quite high for children nearest the smelter. Household dustiness, soil lead, and estimated or observed air lead concentrations were obtained for 9 zones around the smelter and used to calculate daily lead uptake levels. Like the Omaha comparison, dietary lead intakes were calculated from 1970s data analyzed in the criteria document. The geometric mean values by zone are shown in Table 5 along with predicted mean PbB using the uptake/biokinetic model. Results are

Table 5. *Children's blood levels measured in Silver Valley, Montana 1974–1975* versus *uptake/biokinetic model predictions.*

Area	I	II	III	IV	V	VI	VII
Average air lead ($\mu g/m^3$)	16.8	14.2	6.6	3.0	0.7	0.5	0.5
Average soil lead ($\mu g/g$)	7,470	3,300	1,250	1,400	2,300	337	700
Average house dust lead ($\mu g/g$)	11,700	10,300	2,400	3,300	3,400	1,800	3,900
Observed PbB ($\mu g/dL$) (2 year olds)	72	51	36	35	35	25	35
Predicted mean PbB* ($\mu g/g$) (2 year olds)	87.8	73.5	26.8	32.0	33.9	20.6	34.2

Estimated dietary lead uptake (μg/day) for all sites during early 1970s (see text).

0–1 year olds	16.3–20.6
1–2 year olds	20.2–25.6
2–3 year olds	16.6–22.1

*Predicted mean PbB represents average of lower and upper bound means estimated by model.

generally adequate for the more distant areas with lower exposure levels. This is the intended range of applicability of the model. It is not surprising that blood leads are overpredicted in areas I and II located closest to the smelter. The Harley and Kneip compartmental model's presumption of linear kinetics in lead absorption, excretion, and accumulation, while appropriate at relatively low to moderate exposures is probably not applicable for such extraordinarily high exposure levels as observed in these two areas. Some adjustment for the non–linear patterns of lead uptake would be necessary in order reliably to model blood lead levels in locales with such extremely high air, soil, and/or dust lead concentrations.

Additional Studies

Much statistical work is being done on validation of the uptake/biokinetic model. These include tests of predictive validity for an individual child, tests of linear calibration of the model, and tests of the equivalence of the observed and simulated distributions. Results will be reported elsewhere. Some work is also underway on modelling the uptake of lead by the foetus due to maternal lead exposure. The preliminary results (Marcus *et al.*, 1988) apply only to models for lead-exposed rats. Extrapolation to humans and validation tests have not yet been carried out.

Why We Use Linear Models when the Relationship is Nonlinear

There is compelling evidence that the relationship between blood lead and lead uptake is nonlinear. A large amount of effort has been put into fitting nonlinear descriptive models [see Chapter 11 (US EPA, 1986a) for a list of models]. Yet, the biological models of the previous section assume first-order linear pharmacokinetics and absorption, which implies linearity of blood lead with uptake. Can this be resolved? We believe that much of the apparent low-dose nonlinearity cited in the literature is due to statistical artifacts, most particularly omission of important covariates such as dust lead in fitting water lead relationships.

We have re-analyzed the data on PbB *versus* PbW for N = 92 infants in Glasgow published in Lacey *et al.* (1985). The cube root model fitted the data, but other models based on nonlinear kinetics of absorption or blood distribution fitted significantly better with one or two additional adjustable and biologically interpretable parameters (Marcus and Holtzman, 1988). An indication of the potential confounding between dust lead and water lead is shown in Figure 2 in Laxen *et al.* (1987), in which most children with very high water lead had low dust lead exposure (hence lower blood lead), and many children with low water lead had very high dust lead. Since lead is a multimedia pollutant, it is essential that all potentially significant sources of lead uptake be measured before any relationship is fitted to the data. A nonlinear relationship between blood lead and lead exposure is also evident in the Boston data (Rabinowitz *et al.*, 1985) and Cincinnati data (Bornschein *et al.*, 1985) discussed previously. Thus the real questions are: why is the blood lead-soil lead slope nonlinear and, when shall we depart from a linear relation?

Many biological lead processes are nonlinear. For example, gut absorption of lead in everted rat intestines has a saturable active transport mechanism as well as a passive (linear) mechanism (Aungst and Fung, 1980). The real problem may arise because blood lead is itself a nonlinear indicator of uptake. Analysis (Marcus, 1985) of cross-sectional studies of plasma lead *versus* blood lead in Australian workers (DeSilva, 1981) shows that blood lead increases much more slowly than plasma lead at blood lead levels above 40–60 μg/dL. This suggests that the lead- holding capacity of the erythrocyte may be saturable and that whole blood lead concentrations do not adequately estimate the active lead burden at higher concentrations. While no similar data exist for children, we have made a very indirect estimate using the erythrocyte protoporphyrin (EP) *versus* blood lead relationship for a large sample of children in the US NHANES II survey (Marcus and Schwartz, 1987). There is very clear evidence that the EP response in children (hence the implicit plasma lead burden) increases much more steeply for children with iron deficiency than for iron-replete children, with a noticeable nonlinear increase in the model at PbB > 25 μg/dL. We would thus provisionally recommend that linear models be used for risk assessment and standards evaluation in children up to PbB = 25 μg/dL (this includes all lead sources such as air, food, water, soil and dust), but that nonlinear models be used at higher concentrations. More work is clearly needed on this problem too.

Summary and Conclusions

We have reviewed a number of models for blood lead in children as a function of soil lead exposure. There are other sources of lead exposure such as air, food, water, dust, and paint, so that interpretation of soil lead models requires information about all other pathways. Among studies with good control of covariates, the relationship between blood lead and soil lead appears approximately linear at blood lead levels below 25 μg/dL.

The blood lead slope estimate in most studies is about 2 μg/dL per 1,000 μg/g soil lead, but may be as large as 7 μg/dL per 1,000 μg/g. Soil and house dust leads are correlated and similar in effect in these studies. Causal models for children in Boston and Cincinnati have shown a strong component of soil lead in dust lead and, both directly and indirectly, in blood lead.

The linear models are not accurate at high levels of lead intake. There are compelling reasons, both biological and statistical, for including nonlinear processes in models used for PbB $>$ 25 μg/dL. At levels below 15 μg/dL there is equally good reason to use (approximately) a linear model between blood lead and soil lead.

The OAQPS uptake/biokinetic model is a predictive model whose mathematical assumptions and numerical parameters combine plausible biological hypotheses, animal experiments, and results of human observational studies. This model can be used to study the responses to hypothetical scenarios for lead exposure. It is based on linear pharmacokinetics, and extension of the model to very high levels of exposure should include known nonlinear kinetic processes for lead absorption and distribution. The present version of the model has been extensively validated and should be useful for predicting mean blood lead levels under alternative exposure scenarios, including soil lead abatement.

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